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Article type : Review

Handling Editor: Stephanie Januchowski-Hartley

Summary: 275

Main text: 5717

Acknowledgements: 53

References: 2089

Tables: 393

Figure legends: 385

Figure no: 6

Table no: 2

Reference no: **75**

Title: Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit?

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Running title: Eco-engineering marine urban infrastructure

Abstract

1. Along urbanised coastlines, urban infrastructure is increasingly becoming the dominant habitat. These structures are often poor surrogates for natural habitats, and a diversity of eco-engineering approaches have been trialled to enhance their biodiversity, with varying success.

2. We undertook a quantitative meta-analysis and qualitative review of 109 studies to compare the efficacy of common eco-engineering approaches (e.g. increasing texture, crevices, pits, holes, elevations and habitat-forming taxa) in enhancing the biodiversity of key functional groups of organisms, across a variety of habitat settings and spatial scales.

3. All interventions, with one exception, increased the abundance or number of species of one or more of the functional groups considered. Nevertheless, the magnitude of effect varied markedly among groups and habitat settings. In the intertidal, interventions that provided moisture and shade had the greatest effect on the richness of sessile and mobile organisms, while water-retaining features had the greatest effect on the richness of fish. In contrast, in the subtidal, small-scale depressions which provide refuge to new recruits from predators and other environmental stressors such as waves, had higher abundances of sessile organisms while elevated structures had higher numbers and abundances of fish. The taxa that responded most positively to eco-engineering in the intertidal were those whose body size most closely matched the dimensions of the resulting intervention.

4. *Synthesis and applications:* The efficacy of eco-engineering interventions varies among habitat settings and functional groups. This indicates the importance of developing site-specific approaches that match the target taxa and dominant stressors. Furthermore, because different types of intervention are effective at enhancing different groups of organisms, ideally a range of approaches should be applied simultaneously to maximise niche diversity.

Keywords: eco-engineering interventions, artificial structure, crevice, depression, habitat-forming species, microhabitat, protrusion, rockpool, seeding, urban infrastructure

Introduction

Of the many human activities presently contributing to habitat loss and species extinctions, urbanisation is generally considered to have one of the greatest impacts across local to regional scales (Lotze *et al.* 2006; Grimm *et al.* 2008). Over 50% of the human population now lives in urbanised areas (United Nations Population Fund 2007), with areas within 100 km of the coastline particularly heavily developed, housing over 40% of the global population and 60% of its largest cities (>5 million inhabitants, Firth *et al.* 2016a). The urban ecological footprint extends beyond city boundaries and increasingly sprawls into marine and coastal waters (Duarte *et al.* 2008). In addition to introducing pollutants, such as heavy metals, nutrients, artificial light and sound, to marine and coastal habitats (Daoji & Daler 2004; Halpern *et al.* 2008), urban environments introduce infrastructure (Dafforn *et al.* 2015). This infrastructure is used for a range of purposes including coastal protection (e.g. seawalls, breakwaters, groynes), boating or recreational activities (e.g. marinas, piers, pontoons), supply of energy or resources (e.g. oil, gas platforms) and enhancement of fisheries yield (e.g. artificial reefs).

Urban infrastructure impacts on natural ecosystems in a variety of ways, including habitat loss and fragmentation, as well as modification of ecological connectivity, ecosystem functioning and services, and the physico-chemical environment (Fischer & Lindenmayer 2007; McKinney 2008; LaPoint *et al.* 2015; Bishop *et al.* 2017). The net effect is urbanised ecosystems that are fundamentally different in structure and function to the natural habitat which they displace (Airoldi *et al.* 2015; Gittman *et al.* 2016; Heery *et al.* 2017). In some instances the need for urban infrastructure may be circumvented by adding or restoring natural habitats that enhance biodiversity and provide essential functions (Sutton-Grier, Wowk & Bamford 2015; Dethier, Toft & Shipman 2016). For example, the conservation, restoration and/or establishment of coastal plants, and shellfish and coral reefs that dissipate wave energy and stabilise shorelines may prevent the need for revetments and seawalls (Arkema *et al.* 2013) and also enhance fisheries productivity and sequestration of carbon (Barbier *et al.* 2011). In heavily modified environments, conservation and restoration of natural habitats may, however, not be feasible, and novel solutions are required (Hobbs, Higgs & Harris 2009; Lundholm & Richardson 2010). Amongst these, eco-engineering – the inclusion of ecological principles in the design of infrastructure to enhance its ecological value (Bergen, Bolton & Fridley 2001) – can benefit terrestrial and marine environments alike (Chapman & Underwood 2011; Francis & Lorimer 2011). Ideally, ecological values should be incorporated in infrastructure during the design phase to have greatest effect, but existing structures may also be modified to promote species of conservation, commercial or functional interest and to enhance native biodiversity (Chapman & Blockley 2009; Dugan *et al.* 2011).

In terrestrial environments, green walls and roofs have been designed to enhance biodiversity, restore connectivity to certain faunal groups, and bolster desired ecosystem functions (Lundholm & Richardson 2010; Francis & Lorimer 2011; Braaker *et al.* 2014). Analogous approaches can be applied to the design of urban infrastructure in marine environments (Chapman & Underwood 2011; Firth *et al.* 2014a). As compared to the largely horizontal and topographically complex surfaces of natural substrates, marine urban infrastructure typically has vertical, smooth, surface that reduces the area for attachment and the diversity of habitat niches for organisms, and provides fewer refuges from predators, competitors and/or environmental stressors (Bulleri & Chapman 2010; Loke & Todd 2016). Consequently, one of the commonly utilised techniques for eco-engineering marine infrastructure has been to increase surface area and/or habitat complexity of the hard substrate at a range of scales (mm to metres) using either additive (i.e. attachment of protruding structures) or subtractive (i.e. drilling, removal of substrate) processes (Chapman & Underwood 2011). Additive approaches have utilised both abiotic substrate, and ‘seeding’ with habitat-forming taxa such as barnacles, bivalves, canopy-forming algae, branching coralline algae or corals (e.g. Dafforn, Glasby & Johnston 2012; Perkol-Finkel *et al.* 2012; Wilkie, Bishop & O'Connor 2012; Ferse *et al.* 2013). In the marine environment, the majority of eco-engineering to date has been small-scale experimental additions of habitat features to existing urban infrastructures (Chapman & Underwood 2011), with relatively few attempts to incorporate features into new urban infrastructures (but see Chapman & Blockley 2009, Firth *et al.* 2013 for some exceptions). These interventions have had varying degrees of success in enhancing native biodiversity, and in some instances may serve as ecological traps if they lead to organisms utilising habitats that reduce their fitness (Hale, Treml & Swearer 2015; Hale, Morrongiello & Swearer 2016). Despite this, quantitative studies of the factors that influence the efficacy of such interventions in enhancing biodiversity are lacking.

The efficacy of eco-engineering interventions for enhancing the biodiversity of urban infrastructures is likely to vary across species and environments as well as the spatial and temporal scales of the intervention. The stress gradient hypothesis predicts that positive associations will be greatest in environments where biotic or abiotic stressors are greatest, and weakest in environmentally benign environments (Bertness & Callaway 1994). Hence, interventions that ameliorate abiotic stressors such as temperature and desiccation may be expected to have increasingly strong influences across the intertidal gradient (Bateman & Bishop 2017). Interventions that weaken biotic interactions may be most effective in environments with high predator abundances, or in which competition is intense (Chapman & Underwood 2011; Strain *et al.* in review). Additionally, because responses of organisms to complexity are dependent on body size (Hacker & Steneck 1990; McAbendroth *et al.* 2005), an organism may benefit most from an intervention that adds microhabitats that are a similar order of magnitude to its size (Köhler, Hansen & Wahl 1999). The effects of the interventions can also vary through time depending on the recruitment and growth of the organisms, the mobility of the organism and the successional stage of the community (Firth *et al.* 2016a). For example, the effectiveness of some interventions may only become apparent after sufficient time has elapsed for colonisation to occur (Evans 2016). Alternatively, the efficacy of others may plateau over time, where seeding of structures with biogenic habitats speeds up succession but does not change the endpoint after a number of years (Ferse *et al.* 2013). Studies quantifying how the efficacy of these interventions varies across multiple locations, environments, spatial-scales and time points are lacking.

In this study, we used a meta-analysis and a qualitative literature review to assess sources of variation in the efficacy of interventions aimed at enhancing the biodiversity of both new and existing marine urban infrastructure through the creation of novel microhabitats. We expected that across all scales (ranging from mms to 10s of meters), the addition of complex microhabitats (i.e. texture, crevices, pits, water retaining, holes, small elevations, large elevations, seeding) to urban infrastructure would produce an overall positive effect on the number and abundances of species for specific functional groups (sessile, mobile, benthic, fish) and habitat-forming taxa (barnacles, bivalves, branching coralline, canopy algae, coral). Nevertheless, we expected that the magnitude and direction (positive or negative) of effects of interventions on the abundance and richness of taxa would vary between habitat contexts (intertidal and subtidal) across which the identity of dominant stressors varies, through time, between interventions applied to new and existing infrastructure and among functional groups of organisms, reflecting variation in their niche requirements, and body size.

Materials and methods

Literature search

We searched the literature using Google Scholar and Web of Science for manipulative and mensurative field studies in intertidal and subtidal estuarine and coastal marine systems that examined the ecological effects of adding microhabitats to urban infrastructure (i.e. directly to structures or to settlement panels) either during construction or by retrofitting. The search terms included (‘microhabitats*: texture*, roughness* crevices*, cuts*, fissures*, grooves*, pits*, rockpools*, tidal pools*, rock pools* flowerpots* holes*, ridges*, elevations*, towers*, raises*, relief*, mimic*, rope*, ribbons*, brushes*’) and (‘seeding*, transplants*, planting*, epoxy*, glue*, habitat-forming*, barnacles*, bivalves*, mussels*, oysters*, canopy*, kelps*, coral*, branching coralline*, corticated turf*, branching turf*’) on (‘artificial habitat*, artificial reefs*, artificial structure*, tiles* or settlement plates*’). We also searched the reference and citation lists of each article identified using the same search terms.

We selected studies for the analyses that compared between otherwise similar urban infrastructure with and without the intervention: (1) the number of species per unit area (i.e. species density); (2) the abundance of all species within one or more functional groups: sessile algae and invertebrates (hereafter ‘sessile’), mobile invertebrates (hereafter ‘mobile’), all sessile algae and sessile and mobile invertebrates combined (hereafter ‘benthic’) and fish (hereafter ‘fish’); and/or (3) the species density and total abundance of key habitat-forming taxa (see Table 1 examples). For each study, the nature of the intervention was classified according to whether it added texture, crevices, pits, intertidal water retaining features, subtidal holes, elevations, or habitat-forming species (see Table 1 for definitions) to urban infrastructure. For studies that tested the effects of multiple types of intervention or single types of intervention, across multiple sites each intervention and site was used as a replicate for the analyses (see below for further details).

Data extraction

We found 388 studies through the literature search, from which 109 were suitable for inclusion in our meta-analysis (Table S1) after exclusions (i.e. lack of controls, data on single species or a subset of species from a functional group, confounding with other factors, relevant data not presented either in text or graphs). A list of the studies used is provided in the Data sources section. For each study, we recorded the sample size, and the mean and standard deviation (when reported) of the number and/or abundance of each functional group on urban infrastructure receiving the intervention and on otherwise similar unmanipulated substrate (control). In instances where data were presented in the figures, we used GetData Graph Digitizer version 2.25.0.32 (www.getdata-graph-digitzer.com) to extract means and standard deviations. We also recorded the geographical location of each study, the time interval after which the invention was fitted or built (in months; hereafter 'time'), the type of intervention either retrofitted or built (hereafter 'method'), the area across which the intervention was applied (m^2) and the dimensions of the unit of intervention (i.e. depth of crevices, pits, holes, intertidal water retaining features and height of elevations and habitat-forming taxa), where available.

Data analysis

For studies reporting means, standard deviations and sample sizes (or from which these data could be extracted from figures), we calculated the effect size of the various interventions on variables of interest (i.e. abundance and number of species) as Hedge's g standard mean difference (SMD) (Hedges 1981). We chose the SMD effect size in the meta-analysis rather than the log ratio because these data contained many zeros (i.e. no species observed and/or no variance observed between replicates within the same treatment), (Borenstein *et al.* 2010). For the analysis, the effects of interventions were tested against the control using a random effects model as there was significant heterogeneity between studies (determined by measuring heterogeneity via Cochran's Q, and testing it against a v^2 distribution with n-1 degrees of freedom, where n is the number of studies). The model was fitted using the Hedges random effects estimator (Hedges 1981).

For studies that tested the effect of interventions at different sites, we treated each site as a separate study in the meta-analysis. We tested for links between these by adding study identity as a moderator in the model. When sites from the same study were linked, the results were adjusted by adding study identity as a moderator in a multilevel random effects model.

For each functional group and habitat-forming taxa we assessed how the magnitude and direction (positive or negative) of effects varied with the size of the intervention area (m^2), the depth or height of the unit of intervention (either the depression or elevation in mm to m), the time after implementation of the intervention that monitoring was done (months), method (retrofitted or built) and differences between zones (intertidal or subtidal) and the type (Table 1) by adding these terms separately, as moderators in the models. Similarly, for each type of intervention, we assessed how the magnitude and direction of effects varied across the functional groups or habitat-forming taxa by including intervention type (Table 1) as moderators in the models. For the water retaining features, only data on the species number was presented in the studies, and not the species abundances. Therefore, we could not compare the effects of water retaining features on species abundances to the other interventions (i.e. texture, crevices, pits, small elevations, or seeding) in the analyses.

For studies that did not present the variance between replicates, we substituted in the maximum standard deviation from studies on the same intervention (Furukawa *et al.* 2006; Strain *et al.* 2014). There were no detectable differences in effect sizes between the studies with and without standard deviations (based on overlapping 95% confidence intervals). We also tested and found no differences in the effects of the microhabitats between the manipulative (97%) or mensurative (3%) studies (data not shown).

We checked whether there was a significant correlation between the effect size and sample size, as a measure of publication bias using qualitative tests (weighted frequency histogram, funnel plots and Q–Q normality plots of effect sizes). We also assessed the number of studies required to increase the p-value to above 0.05, using the Rosenthal's fail-safe number test (Tables S2-3). All analyses and plots were undertaken using the R package, *metafor* (Viechtbauer 2010) in R gui 3.1.1 (R Core Team 2016).

In addition, we undertook a qualitative review that included studies that did not present data that could be extracted for the analysis (i.e. only written statements about their results). For each type of intervention, we calculated the proportion of studies reporting significant versus non-significant results. We tested for differences in the proportion of significant studies between intertidal and subtidal zones, or among functional groups or habitat-forming taxa using $\chi 2$ proportions tests.

For both the overall meta-analysis and qualitative review, we used the data from the final sampling period of each study. We only performed analyses on interventions with three or more studies (Tables S2-6).

Results

Of the 109 studies from which data were extracted, 23% focused on texture and 21% on crevices. The remaining studies, focused on pits, water retaining features, subtidal holes, small elevations, large elevations and seeding, each contributed between 3-12% to the total number of studies used in the review. 67% of studies described interventions that were retrofitted to existing structures, with the remainder describing interventions that were incorporated at the design stage (Table S1). Of the studies describing interventions at the design stage, 72% were on artificial reefs (Table S1). The studies were not evenly distributed around the globe (Fig. 1) and much (60%) of the research was conducted in Australia (Sydney), Israel (Red Sea), Europe (various locations) and North America (east coast).

The studies were published between 1946 and 2016, with a sharp increase in number through time, mainly between 1990 and 2016 (Fig. 2). This trend is likely to be driven in part by the increasing urbanisation of marine coastlines across the globe and the strong associated interest in eco-engineering approaches. Each intervention type had studies from multiple laboratories, years and countries, indicating the review conclusions are not strongly biased towards an individual country or time point (Table S1).

Most types of intervention (all but the addition of large elevations) significantly enhanced the number and/or abundance of species for at least one key functional group and/or habitat-forming taxon relative to the control (Figs. 3, 4, 5; Tables S2-S6). Interestingly, in only one instance - the addition of texture to the subtidal – was the abundance of a group (the barnacles) significantly reduced relative to the control (Figs. 3-5). The most effective interventions in increasing the number of species were water retaining features (mean [±SE] difference for sessile and benthic species = 5.0 ±4.4) and intertidal pits (mean [±SE] difference for benthic species = 4.7 ±2.1) and to a lesser extent intertidal crevices (mean [±SE] sessile species = 2.2 ±1.6), and subtidal soft interventions (mean [±SE] difference in fish species = 1.6 ±2.0) and seeding (mean [±SE] difference in sessile and fish species = 2.4 ±2.8). There were no detectable differences in effects of retrofitted or built interventions on the number or abundances of species, so these methods were pooled for the final analyses (Tables S2-S3).

For many of the interventions (texture, crevices, pits, subtidal holes, small elevations and large elevations, soft structures and seeding), the area of the intervention had a weak non-significant positive effect on the number of species (Table S2-S3). In contrast, for intertidal water retaining features, there was a significant positive effect of intervention area on the number of species for each of the functional groups (Table S2). There was no relationship between area of intervention and abundances of species for any of the interventions (Tables S2-S3). As predicted, the effect of most of the interventions (texture, crevices, pits) differed between zones (Tables S2-S6). In contrast, there were no clear effects of the height or depth of the unit of manipulation (i.e. depression or elevation), or the time (months) of the intervention on the species number or abundance (Tables S2-S3).

Overall the results from the meta-analysis and the qualitative review showed similar trends (Table 2). For each intervention we highlight the results of the meta-analysis where available and the results from the qualitative review where there was insufficient information presented to undertake the meta-analysis.

Effect of intervention type on the number and abundances of species by functional group
The efficacy of the interventions in enhancing the species number and abundance of key functional groups varied among categories (Figs. 3, 4; Table S4-S6). For sessile organisms, the meta-analysis demonstrated that crevices, water retaining features, or seeding in the intertidal zone resulted in greater increases in the number of species than any of the other interventions tested, in either the intertidal or subtidal zone (Q_4 = 40.0, p <0.001, Fig. 3, Table S2). In contrast, the cover of sessile species displayed a greater positive response to intertidal seeding and the addition of subtidal texture than to the other interventions (Q_3 = 8.3, p = 0.049, Fig. 4, Table S3). For the mobile species, the qualitative review found that a greater proportion of studies displayed significant effects of intertidal crevices, pits or subtidal holes on abundances ($\chi 2_3$ = 10.4, p = 0.015) but not numbers of species ($\chi 2_3$ = 7.3, p

> 0.05), relative to the other interventions (Figs. 3, 4; Tables S4-S5). For fish, the meta-analysis suggested subtidal soft features and seeding were most important for enhancing both the number ($Q_4 = 36.0$, $p < 0.001$) and abundances of species ($Q_4 = 15.6$, $p = 0.004$) relative to the other interventions tested (Figs. 3, 4; Tables S2-S3). As expected, the qualitative analysis also showed that in a greater proportion of studies, intertidal water retaining features enhanced the number of fish species as compared to the other interventions assessed ($\chi 2_4 = 12.7$, $p = 0.013$; Fig. 3; Table S4).

Across the different interventions, intertidal water retaining features and seeding (irrespective of zone) were the only habitats that significantly enhanced the number of species for multiple functional groups (Figs. 3, 4; Table S4). The meta-analysis demonstrated that intertidal water retaining features significantly increased the number of sessile, benthic and fish species, but not mobile species relative to controls ($Q_3 = 9.2$, $p = 0.036$, Fig. 3, Table S2). Seeding resulted in a significantly higher number ($Q_4 = 13.4$, $p = 0.009$) and abundance ($Q_4 = 36.8$, $p < 0.001$) of intertidal sessile species and subtidal fish but not intertidal mobile species or subtidal sessile species (Figs. 3, 4; Tables S2-S3).

In contrast, the addition of texture, crevices, pits, subtidal holes or soft structures to urban infrastructure only enhanced the species number or abundance of a single functional group (Figs. 3, 4; Tables 1, S2). The meta-analysis showed the addition of subtidal texture only significantly enhanced the cover of sessile species (Figs. 3, 4; Table S2). Intertidal crevices increased the number of intertidal sessile species and pits increased the number of benthic species, but the qualitative analyses suggested both of these interventions in many studies also resulted in higher abundances of mobile species (Figs. 3, 4; Tables S2, S4). Subtidal holes only significantly increased the abundances of mobile species (Figs. 3, 4; Tables 1, S3), while the addition of soft habitats significantly increased the number and abundances of fish species (Figs. 3, 4; Tables 1, S2-S3).

Effect of intervention type on the number and abundance of habitat-forming taxa

As predicted, many of the interventions significantly increased the abundance of habitat-forming taxa (Fig. 5; Tables S3, S6). For barnacles ($Q_7 = 7.8$, $p = 0.049$) and bivalves ($Q_6 = 8.8$, $p = 0.048$), the meta-analysis showed the addition of intertidal crevices and pits resulted in higher cover and/or counts relative to the other interventions tested (Fig. 5; Tables S3). In contrast, for corals, the addition of subtidal pits had the greatest benefits of all the interventions considered ($Q_2 = 10.5$, $p = 0.006$; Fig. 5; Tables S3). The qualitative analysis also showed in a greater proportion of studies the addition of texture resulted in increased cover of branching coralline ($\chi 2_5 = 18.0$, $p = 0.003$; Fig. 5; Tables S6), while small elevations lead to higher cover of canopy-forming algae ($\chi 2_5 = 18.0$, $p = 0.003$, Fig. 5; Table S6) relative to the other interventions tested.

Overall, the addition of pits had the greatest benefits for multiple groups of habitat-forming taxa (Fig. 5; Tables S3, S5). The meta-analysis showed intertidal pits significantly increased the abundances of barnacles and bivalves ($Q_5 = 88.7$, $p <0.001$, Fig. 5, Table S3). The qualitative review suggested this intervention could also lead to higher cover of branching coralline algae while subtidal pits significantly increased the cover or counts of barnacles, branching coralline algae and corals (Fig. 5; Table S6). The addition of texture to the intertidal resulted in significantly higher counts and cover of barnacles, branching coralline and slightly more bivalves and in the subtidal increased cover of branching coralline algae, but there were no detectable effects of this intervention on the other taxa ($Q_7 = 30.7$, $p <0.001$; Fig. 5; Table S3). Crevices had significantly higher counts of barnacles and cover of bivalves when situated in the intertidal, but there were no detectable effects of this intervention on other intertidal taxa or in the subtidal ($Q_5 = 25.0$, $p <0.001$; Fig. 5; Table S3, S5). The qualitative analysis showed that a greater proportion of studies demonstrated intertidal water retaining features resulted in significantly higher numbers of species of branching coralline and canopy-forming algae ($\chi 2_2 = 11.9$, $p = 0.008$; Fig. 5; Table S6), and small elevations increased the cover of intertidal canopy-forming algae ($\chi 2_2 = 5.6$, $p = 0.049$; Fig. 5, Table S6) relative to the other interventions. Interestingly, there were no clear benefits of seeding on the abundances of new recruits of bivalves, coral or canopy-forming algae ($Q_3 = 2.4$, $p >0.05$; Fig. 5; Table S3).

Discussion

The effective use of eco-engineering as a tool for enhancing the habitat value of urban infrastructure requires knowledge of when and where interventions have greatest influence. Despite this, most eco-engineering studies in marine environments have focused on a single type of microhabitat-enhancing intervention, at one or few sites (e.g. Chapman & Blockley 2009; Browne & Chapman 2014; Firth *et al.* 2014a). Studies in natural systems demonstrate how the responses of species assemblages to microhabitats can vary across environmental gradients (e.g. Firth *et al.* 2014b; McAfee, Cole & Bishop 2016) and among taxa (e.g.

Bateman & Bishop 2017). Our study provides the first cross-study, quantitative assessment of how the effectiveness of different interventions applied to marine urban infrastructure varies among groups of organisms and environmental settings. As predicted (see reviews by Dafforn *et al.* 2015; Dyson & Yocom 2015; Firth *et al.* 2016a), overall microhabitat-enhancing interventions had a positive effect on the abundance and number of species across the studies. Nevertheless, the magnitude of their effects varied considerably, from zero to highly positive according to the type of intervention, the target taxa, and tidal elevation.

In the intertidal, thermal and desiccation stresses have long been implicated in setting distributional limits (e.g. Wolcott 1973; Harley 2003) and the persistence of organisms can be contingent on the availability of microhabitat refugia from such stressors (Silliman *et al.* 2011; Firth *et al.* 2016b; McAfee, Cole & Bishop 2016). Perhaps not surprisingly then, the intertidal interventions with the largest influence on sessile organisms, including barnacles, bivalves, branching coralline and canopy-forming algae, and on mobile organisms, were crevices, pits, and water retaining features, each of which provide shading and moisture retention at low tide (Fig. 6, Table 5, Garrity 1984; Underwood & Jernakoff 1984). Similarly, fish, which in the absence of water retaining features cannot persist in the intertidal zone at low tide, were strongly influenced by water-retaining interventions. In contrast, the addition of small elevations had little, if any, effect on intertidal organisms, despite their capacity to enhance surface area for attachment. In the intertidal, the groups of organisms that responded most strongly to a particular type of intervention were those whose body size most closely matched the dimensions of the unit of intervention (Fig. 6, Hacker & Steneck 1990; McAbendroth *et al.* 2005). For example, small-scale enhancements, such as adding texture, pits and crevices, were most effective for smaller bodied organisms such as barnacles and bivalves. In contrast, larger interventions such as rock pools could also support larger species such as branching coralline, canopy-forming algae and fish (Fig. 6).

Similarly, subtidal interventions that added depressions, as opposed to elevations, generally had greatest positive effects on the majority of taxa. Whereas in the intertidal such interventions serve to retain moisture, in the subtidal they may be more important in providing refuge from large-bodied predators, such as fish, which can exert considerable top-down control on the biota on marine infrastructure (Connell & Anderson 1999; Clynick, Chapman & Underwood 2007; Ferrario *et al.* 2016). Depressions can also serve as protection from high wave exposure that can challenge the attachment strength of organisms and interfere with feeding behaviour (Moschella *et al.* 2005; Bulleri & Chapman 2010). In contrast, the elevated structures formed by seeding marine infrastructure with large-bodied habitat-forming taxa or soft structures (e.g. rope) had greater positive influence on subtidal fish than depressions. Such larger-bodied taxa may not fit within the bounds of depressions, and instead, elevated structures may provide shelter and food resources for these (Hair & Bell 1992; Fernández *et al.* 2009). However, in the subtidal, a relationship between the body-size of organisms and the dimensions of the interventions that produced the most positive effect sizes was not demonstrated (Fig. 6).

Although most of the eco-engineering interventions that we reviewed manipulated microhabitats through the addition and/or subtraction of abiotic habitat, approaches that add biotic microhabitat through seeding with habitat-forming species may serve to provide additional benefits (Dafforn *et al.* 2015). Not only may such interventions add habitat, and mitigate the effect of abiotic and biotic stressors on associated organisms (Dafforn, Glasby & Johnston 2012), but they may also play an important role in carbon sequestration (e.g. macroalgae), nutrient cycling and/or maintain clean waters (e.g. filter feeders). Nevertheless, the establishment of habitat-forming taxa remains a challenge on some urban infrastructure (Bulleri & Chapman 2010). For example, while transplant of the canopy-forming algae *Cystoseira barbata* onto breakwaters is technically feasible, survivorship can be limited by grazing, which is more intense than on natural rocky reefs (Perkol-Finkel *et al.* 2012; Ferrario *et al.* 2016). Additionally, because the location of infrastructure is often in areas that suffer from high pollutant loadings and poor water quality, environmental conditions may limit the growth and survivorship of habitat-forming species (Falace, Zanelli & Bressan 2006; Ng *et al.* 2015).

Although our meta-analysis demonstrated predominantly positive effects of microhabitat interventions on the abundance and number of species of key functional groups of organisms, very few of the studies identified and analysed, provided assessment of the proportion of species that were native, non-native or cryptogenic (of unknown origin; e.g. Dafforn, Glasby & Johnston 2012; Sella & Perkol-Finkel 2015). In highly urbanised environments, with a long history of shipping and exploitation, the high proportion of species that are cryptogenic can complicate such assessments (Bishop & Hutchings 2011). Nevertheless, despite such difficulties, a large body of literature suggests that subtidal urban infrastructures support more non-native species than nearby rocky reefs (Dafforn, Glasby & Johnston 2012; Airoldi *et al.* 2015) and sedimentary habitats (Heery *et al.* 2017). Assessing the extent to which native, non-native and cryptogenic species benefit from interventions would help to identify maladaptive scenarios which lead to proliferation of unwanted pest species, as well as approaches that limit such risk. For example, interventions that manipulate microhabitat through the addition of biotic (i.e. habitat-forming species) as opposed to abiotic structure, may lessen risk of rapidly colonising pest species from dominating structures, by pre-empting space that they may otherwise occupy (Dafforn, Glasby & Johnston 2012).

Our analysis revealed that the majority of eco-engineering interventions involved patch-scale, short-term manipulations of individual microhabitat types. These small-scale interventions do not recreate the properties of contiguous natural habitats, due to their comparatively large edge to interior ratios and small areas (Bender, Contreras & Fahrig 1998). Interventions at the scale of the entire structure remain rare, and consequently, our knowledge of how biodiversity benefits relate to the scale of the infrastructure remains poor. As some mobile species, such as grazers or fish, might require a minimum habitat area in order to effectively forage (Perkins *et al.* 2015), it is expected that a positive relationship between the area of interventions and their effect on biodiversity might emerge as larger-scale interventions are attempted. Additionally, because the majority of monitoring associated with such interventions was also at the patch scale, and rarely extended beyond 12 months (but see Ferse *et al.* 2013) our understanding whether such eco-engineering approaches have biodiversity benefits that extend beyond the site of the intervention or over longer timeframes remains largely unknown. None of the studies tested the benefits of providing habitat complexity at multiple scales.

The studies assessing the efficacy of eco-engineering interventions came primarily from developed countries in North America, Europe and Australasia. Although this may be a function of both the distribution of coastal ecologists monitoring eco-engineering interventions, and the distribution of eco-engineering interventions themselves, we suspect that the latter is the key driver of this non-random distribution. In terrestrial environments, socioeconomic status is a key indicator of the uptake of eco-engineering interventions such as green walls and roofs, which correlates with factors such as level of education, willingness to pay for environmental improvements, and the resources available for creating an ecological ideal (Kinzig *et al.* 2005; Francis & Lorimer 2011). While such studies are not yet available for the marine environment, we expect similar drivers for the uptake of marine eco-engineering. Quantification of the economic benefits of marine eco-engineering interventions relative to any additional costs associated with their incorporation into structures would help to increase the support for broader-scale implementation.

While the eco-engineering of marine urban infrastructure has made significant advances in the past few decades, there has been little consideration of how specific local scale abiotic factors (e.g. pollution, temperature, wave exposure) or biotic interactions (e.g. predation, competition, facilitation) influence species interactions and distributions (Bulleri & Chapman 2010). This is despite predictions of ecological theory that positive interactions will strengthen across gradients of biotic (e.g., competition, predation, facilitation) and/or abiotic (e.g., temperature, desiccation) stress, while negative interactions will weaken (Bertness & Callaway 1994). Although our review clearly shows that the effects of complex microhabitats are generally positive, the differing effect size of many of the interventions between intertidal and subtidal zones, and between groups of species, highlights the important role that interactions with the environment can play in determining the outcome of eco-engineering.

The goals of eco-engineering may range from enhancement of biodiversity, to enhancement of specific ecosystem services, such as fisheries productivity, carbon sequestration, maintenance of water clarity and/or nutrient cycling (Chapman & Underwood 2011). The results of this meta-analysis will assist managers and stakeholders in identifying solutions that best match their specific goals. As different groups of organisms responded most strongly to different types of intervention, eco-engineering projects aimed at maximising biodiversity might benefit from the creation of a variety of different types of microhabitats on any given structure, that increase the breadth of niche space available to organisms (Connor & McCoy 1979). In contrast, projects aimed at enhancing fisheries productivity may wish to target those interventions - the addition of water-retaining features to the intertidal or habitat-forming species or structural mimics to the subtidal – that maximise fish abundance. Nevertheless, studies examining the efficacy of eco-engineering interventions in enhancing ecosystem services are rare, and only one study (Loke & Todd 2016) has tested the effects of utilising mosaics of multiple types of interventions. However, this study did not quantify the benefits of adding a mosaic of interventions vs. individual interventions for enhancing the richness for multiple functional groups or habitat-forming taxa (Loke & Todd 2016). Further research is urgently needed on these topics. Recent advances in computation design software and three-dimensional printing technology now allow for bespoke eco-engineering designs to be cheaply and readily developed for individual sites (Loke *et al.* 2014). Such techniques also offer great potential for re-creating structures/surfaces that are more akin to natural shorelines.

Although the results of this study indicate that eco-engineering interventions enhance the abundance and richness of ecological communities associated with urban infrastructure, it is unclear to what extent these interventions mitigate the impact of replacing natural with artificial habitat. In addition to local-scale impacts on biodiversity, urban infrastructure can impact ecological processes over larger scales by modifying ecological connectivity (Bishop *et al.* 2017) and through the cumulative effects of multiple developments (Dethier, Toft & Shipman 2016). Given that eco-engineering interventions are unlikely to fully compensate for impacts of urban infrastructure, the feasibility of 'nature-based' approaches, which entail restoration, conservation or creation of habitats that provide the desired functions of infrastructure, should first be investigated prior to the decision to build new structures (Sutton-Grier, Wowk & Bamford 2015; Dethier, Toft & Shipman 2016). Where it is not possible to avoid the construction or removal of infrastructure, eco-engineering approaches, which are mindful of site characteristics, the local species pool, and project goals, can assist in minimising the ecological footprint.

Acknowledgements

This study was part of the World Harbour Project, funded by the Ian Potter Foundation, the Harding Miller Foundation and NSW Government Office of Science and Research. MJB received funding from the NSW Office of Environment and Heritage through the Coastal Processes and Responses Node of the NSW Adaptation Hub. We thank the Editor and reviewers for their insightful comments an earlier draft of the study.

Authors' contributions

ES and MB conceived the ideas, designed methodology and led the writing of the manuscript. ES analysed the data. All authors collected the data, contributed critically to the drafts and gave final approval for publication.

Data accessibility

A list of data sources used in the study is provided in the Data sources section. Further information is also available in Table S1 in online Supporting Information.

References:

Airoldi, L., Turon, X., Perkol-Finkel, S. & Rius, M. (2015) Corridors for aliens but not for natives: effects of marine urban sprawl at a regional scale. *Diversity and Distributions,* **21,** 755-768.

Arkema, K.K., Guannel, G., Verutes, G., Wood, S.A., Guerry, A., Ruckelshaus, M., Kareiva, P., Lacayo, M. & Silver, J.M. (2013) Coastal habitats shield people and property from sea-level rise and storms. *Nature Climate Change,* **3,** 913-918.

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C. & Silliman, B.R. (2011) The value of estuarine and coastal ecosystem services. *Ecological Monographs,* **81,** 169-193.

Bateman, D. & Bishop, M.J. (2017) The environmental context and traits of habitat-forming bivalves influence the magnitude of their ecosystem engineering. *Marine Ecology Progress Series*. **563,** 95-110.

Bender, D.J., Contreras, T.A. & Fahrig, L. (1998) Habitat loss and population decline: a meta-analysis of the patch size effect. *Ecology,* **79,** 517-533.

Bergen, S.D., Bolton, S.M. & Fridley, J.L. (2001) Design principles for ecological engineering. *Ecological Engineering,* **18,** 201-210.

Bertness, M.D. & Callaway, R. (1994) Positive interactions in communities. *Trends in Ecology and Evolution,* **9,** 191-193.

Bishop, M. & Hutchings, P. (2011) How useful are port surveys focused on target pest identification for exotic species management? *Marine Pollution Bulletin,* **62,** 36-42.

Bishop, M.J., Mayer-Pinto, M., Airoldi, L., Firth, L.B., Morris, R.L., Loke, L.H.L., Hawkins, S.J., Naylor, L.A., Coleman, R.A., Chee, S.Y. & Dafforn, K.A. (2017) Effects of ocean sprawl on ecological connectivity: impacts and solutions. *Journal of Experimental Marine Biology and Ecology*.

Borenstein, M., Hedges, L.V., Higgins, J. & Rothstein, H.R. (2010) A basic introduction to fixed-effect and random-effects models for meta-analysis. *Research Synthesis Methods,* **1,** 97-111.

Braaker, S., Ghazoul, J., Obrist, M. & Moretti, M. (2014) Habitat connectivity shapes urban arthropod communities: the key role of green roofs. *Ecology,* **95,** 1010-1021.

Browne, M.A. & Chapman, M.G. (2014) Mitigating against the loss of species by adding artificial intertidal pools to existing seawalls. *Marine Ecology Progress Series,* **497,** 119-129.

Bulleri, F. & Chapman, M.G. (2010) The introduction of coastal infrastructure as a driver of change in marine environments. *Journal of Applied Ecology,* **47,** 26-35.

Castleton, H., Stovin, V., Beck, S. & Davison, J. (2010) Green roofs; building energy savings and the potential for retrofit. *Energy and Buildings,* **42,** 1582-1591.

Chapman, M.G. & Blockley, D.J. (2009) Engineering novel habitats on urban infrastructure to increase intertidal biodiversity. *Oecologia,* **161,** 625-635.

Chapman, M.G. & Underwood, A.J. (2011) Evaluation of ecological engineering of "armoured" shorelines to improve their value as habitat. *Journal of Experimental Marine Biology and Ecology,* **400,** 302-313.

Clynick, B., Chapman, M. & Underwood, A. (2007) Effects of epibiota on assemblages of fish associated with urban structures. *Marine Ecology Progress Series,* **332,** 201-210.

Connell, S. & Anderson, M. (1999) Predation by fish on assemblages of intertidal epibiota: effects of predator size and patch size. *Journal of Experimental Marine Biology and Ecology,* **241,** 15-29.

Connor, E.F. & McCoy, E.D. (1979) The statistics and biology of the species-area relationship. *The American Naturalist,* **113,** 791-833.

Dafforn, K.A., Glasby, T.M., Airoldi, L., Rivero, N.K., Mayer-Pinto, M. & Johnston, E.L. (2015) Marine urbanization: an ecological framework for designing multifunctional artificial structures. *Frontiers in Ecology and the Environment,* **13,** 82-90.

Dafforn, K.A., Glasby, T.M. & Johnston, E.L. (2012) Comparing the invasibility of experimental reefs with field observations of natural reefs and artificial structures. *PLoS One,* **7,** e38124.

Daoji, L. & Daler, D. (2004) Ocean pollution from land-based sources: East China Sea, China. *AMBIO: A Journal of the Human Environment,* **33,** 107-113.

Dethier, M.N., Toft, J.D. & Shipman, H. (2016) Shoreline Armoring in an Inland Sea: Science-Based Recommendations for Policy Implementation. *Conservation Letters*.

Duarte, C.M., Dennison, W.C., Orth, R.J. & Carruthers, T.J. (2008) The charisma of coastal ecosystems: addressing the imbalance. *Estuaries and Coasts,* **31,** 233-238.

Dugan, J.E., Airoldi, L., Chapman, M.G., Walker, S.J., Schlacher, T., Wolanski, E. & McLusky, D. (2011) 8.02 - Estuarine and coastal structures: environmental effects, a focus on shore and nearshore structures. *Treatise on Estuarine and Coastal Science,* pp. 17-41. Academic Press, Waltham.

Dyson, K. & Yocom, K. (2015) Ecological design for urban waterfronts. *Urban Ecosystems,* **18,** 189-208.

Evans, A.J. (2016) Artificial coastal defence structures as surrogate habitats for natural rocky shores: giving nature a helping hand. PhD, Aberystwyth University.

Falace, A., Zanelli, E. & Bressan, G. (2006) Algal transplantation as a potential tool for artificial reef management and environmental mitigation. *Bulletin of Marine Science,* **78,** 161-166.

Fernández, T.V., D'Anna, G., Badalamenti, F. & Pérez-Ruzafa, A. (2009) Effect of simulated macroalgae on the fish assemblage associated with a temperate reef system. *Journal of Experimental Marine Biology and Ecology,* **376,** 7-16.

Ferrario, F., Iveša, L., Jaklin, A., Perkol-Finkel, S. & Airoldi, L. (2016) The overlooked role of biotic factors in controlling the ecological performance of artificial marine habitats. *Journal of Applied Ecology,* **53,** 16-24.

Ferse, S.C.A., Nugues, M.M., Romatzki, S.B.C. & Kunzmann, A. (2013) Examining the use of mass transplantation of brooding and spawning corals to support natural coral recruitment in Sulawesi/Indonesia. *Restoration Ecology,* **21,** 745-754.

Firth, L.B., Thompson, R.C., Bohn, K., Abbiati, M., Airoldi, L., Bouma, T.J., Bozzeda, F., Ceccherelli, V.U., Colangelo, M.A., Evans, A., Ferrario, F., Hanley, M.E., Hinz, H., Hoggart, S.P.G., Jackson, J.E., Moore, P., Morgan, E.H., Perkol-Finkel, S., Skov, M.W., Strain, E.M., van Belzen, J. & Hawkins, S.J. (2014a) Between a rock and a hard place: Environmental and engineering considerations when designing coastal defence structures. *Coastal Engineering,* **87,** 122-135.

Firth, L.B., Schofield, M., White, F.J., Skov, M.W. & Hawkins, S.J. (2014b) Biodiversity in intertidal rock pools: Informing engineering criteria for artificial habitat enhancement in the built environment. *Marine Environmental Research,* 102, 122-130.

Firth L.B., Knights, A.M., Thompson, R.C., Mieszkowska, N., Bridger, D., Evans, A., Moore, P.J., O'Connor, N.E., Sheehan, E.V. & Hawkins, S.J. (2016a) Ocean sprawl: challenges and opportunities for biodiversity management in a changing world. *Oceanography and Marine Biology Annual Reviews*. 54, 193-269

Firth, L.B., Browne, K.A., Knights, A.M., Hawkins, S.J. & Nash, R. (2016b) Eco-engineered rock pools: a concrete solution to biodiversity loss and urban sprawl in the marine environment. *Environmental Research Letters,* **11,** 094015.

Firth, L.B., White, F.J., Schofield, M., Hanley, M.E., Burrows, M.T., Thompson, R.C., Skov, M.W., Evans, A.J., Moore, P.J. & Hawkins, S.J. (2016c) Facing the future: the importance of substratum features for ecological engineering of artificial habitats in the rocky intertidal. *Marine and Freshwater Research,* **67,** 131-143.

Fischer, J. & Lindenmayer, D.B. (2007) Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography,* **16,** 265-280.

Francis, R.A. & Lorimer, J. (2011) Urban reconciliation ecology: the potential of living roofs and walls. *Journal of Environmental Management,* **92,** 1429-1437.

Furukawa, T.A., Barbui, C., Cipriani, A., Brambilla, P. & Watanabe, N. (2006) Imputing missing standard deviations in meta-analyses can provide accurate results. *Journal of Clinical Epidemiology,* **59,** 7-10.

Garrity, S.D. (1984) Some adaptations of gastropods to physical stress on a tropical rocky shore. *Ecology,* **65,** 559-574.

Gittman, R.K., Scyphers, S.B., Smith, C.S., Neylan, I.P. & Grabowski, J.H. (2016) Ecological consequences of shoreline hardening: a meta-analysis. *Bioscience***,** biw091.

Goff, M. (2010) Evaluating habitat enhancements of an urban intertidal seawall: Ecological responses and management implications. MSc, University of Washington.

Grimm, N.B., Faeth, S.H., Golubiewski, N.E., Redman, C.L., Wu, J., Bai, X. & Briggs, J.M. (2008) Global change and the ecology of cities. *Science*, **319,** 756-760.

Hacker, S.D. & Steneck, R.S. (1990) Habitat Architecture and the Abundance and Body-Size-Dependent Habitat Selection of a Phytal Amphipod. *Ecology,* **71,** 2269-2285.

Hair, C. & Bell, J. (1992) Effects of enhancing pontoons on abundance of fish: initial experiments in estuaries. *Bulletin of Marine Science,* **51,** 30-36.

Hale, R., Morrongiello, J.R. & Swearer, S.E. (2016) Evolutionary traps and range shifts in a rapidly changing world. *Biology Letters,* **12,** 20160003.

Hale, R., Treml, E.A. & Swearer, S.E. (2015) Evaluating the metapopulation consequences of ecological traps. *Proceedings of the Royal Society of London B: Biological Sciences,* **282,** 20142930.

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C. & Fox, H.E. (2008) A global map of human impact on marine ecosystems. *Science,* **319,** 948-952.

Harley, C.D. (2003) Abiotic stress and herbivory interact to set range limits across a two-dimensional stress gradient. *Ecology,* **84,** 1477-1488.

Hedges, L.V. (1981) Distribution theory for Glass's estimator of effect size and related estimators. *Journal of Educational and Behavioral Statistics,* **6,** 107-128.

Heery, E., Bishop, M., Critchley, L., Bugnot, A., Airoldi, L., Mayer-Pinto, M., Sheehan, E., Coleman, R., Loke, L. & Johnston, E. (2017) Identifying the consequences of ocean sprawl for sedimentary habitats. *Journal of Experimental Marine Biology and Ecology*.

Hobbs, R.J., Higgs, E. & Harris, J.A. (2009) Novel ecosystems: implications for conservation and restoration. *Trends in Ecology and Evolution,* **24,** 599-605.

Kinzig, A., Warren, P., Martin, C., Hope, D. & Katti, M. (2005) The effects of human socioeconomic status and cultural characteristics on urban patterns of biodiversity. *Ecology and Society,* **10,** 23.

Köhler, J., Hansen, P. & Wahl, M. (1999) Colonization patterns at the substratum-water interface: How does surface microtopography influence recruitment patterns of sessile organisms? *Biofouling,* **14,** 237-248.

LaPoint, S., Balkenhol, N., Hale, J., Sadler, J. & Ree, R. (2015) Ecological connectivity research in urban areas. *Functional Ecology,* **29,** 868-878.

Loke, L.H., Jachowski, N.R., Bouma, T.J., Ladle, R.J. & Todd, P.A. (2014) Complexity for Artificial Substrates (CASU): software for creating and visualising habitat complexity. *PLoS One,* **9,** e87990.

Loke, L.H. & Todd, P.A. (2016) Structural complexity and component type increase intertidal biodiversity independently of area. *Ecology,* **97,** 383-393.

Lotze, H.K., Lenihan, H.S., Bourque, B.J., Bradbury, R.H., Cooke, R.G., Kay, M.C., Kidwell, S.M., Kirby, M.X., Peterson, C.H. & Jackson, J.B. (2006) Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science,* **312,** 1806-1809.

Lundholm, J.T. & Richardson, P.J. (2010) Mini-Review: Habitat analogues for reconciliation ecology in urban and industrial environments. *Journal of Applied Ecology,* **47,** 966-975.

McAbendroth, L., Ramsay, P., Foggo, A., Rundle, S. & Bilton, D. (2005) Does macrophyte fractal complexity drive invertebrate diversity, biomass and body size distributions? *Oikos,* **111,** 279-290.

McAfee, D., Cole, V.J. & Bishop, M.J. (2016) Latitudinal gradients in ecosystem engineering by oysters vary across habitats. *Ecology,* **97,** 929-939.

McKinney, M.L. (2008) Effects of urbanization on species richness: a review of plants and animals. *Urban Ecosystems,* **11,** 161-176.

Moschella, P., Abbiati, M., Åberg, P., Airoldi, L., Anderson, J., Bacchiocchi, F., Bulleri, F., Dinesen, G.E., Frost, M. & Gacia, E. (2005) Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. *Coastal Engineering,* **52,** 1053-1071.

Ng, C.S.L., Lim, S.C., Ong, J.Y., Teo, L.M.S., Chou, L.M., Chua, K.E. & Tan, K.S. (2015) Enhancing the biodiversity of coastal defence structures: transplantation of nursery-reared reef biota onto intertidal seawalls. *Ecological Engineering,* **82,** 480-486.

Perkol-Finkel, S., Ferrario, F., Nicotera, V. & Airoldi, L. (2012) Conservation challenges in urban seascapes: promoting the growth of threatened species on coastal infrastructures. *Journal of Applied Ecology,* **49,** 1457-1466.

R Core Team (2016) R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Sella, I. & Perkol-Finkel, S. (2015) Blue is the new green-ecological enhancement of concrete based coastal and marine infrastructure. *Ecological Engineering,* **84,** 260-272.

Silliman, B.R., Bertness, M.D., Altieri, A.H., Griffin, J.N., Bazterrica, M.C., Hidalgo, F.J., Crain, C.M. & Reyna, M.V. (2011) Whole-community facilitation regulates biodiversity on Patagonian rocky shores. *PLoS One,* **6,** e24502.

Strain, E., Thomson, R.J., Micheli, F., Mancuso, F.P. & Airoldi, L. (2014) Identifying the interacting roles of stressors in driving the global loss of canopy-forming to mat-forming algae in marine ecosystems. *Global Change Biology,* **20,** 3300-3312.

Strain, E.M., Morris, R.L., Coleman, R.A., Figueira, W., Steinberg, P., Johnston, E. & Bishop, M.J. (in review) Increasing microhabitat complexity on seawalls can reduce fish predation on native oysters. *Ecological Engineering*.

Sutton-Grier, A.E., Wowk, K. & Bamford, H. (2015) Future of our coasts: the potential for natural and hybrid infrastructure to enhance the resilience of our coastal communities, economies and ecosystems. *Environmental Science & Policy,* **51,** 137-148.

Underwood, A. & Jernakoff, P. (1984) The effects of tidal height, wave-exposure, seasonality and rock-pools on grazing and the distribution of intertidal macroalgae in New South Wales. *Journal of Experimental Marine Biology and Ecology,* **75,** 71-96.

United Nations Population Fund (2007) State of the world population 2007: unleashing the potential of urban growth. *New York*.

Viechtbauer, W. (2010) Conducting meta-analyses in R with the metafor package. *Journal of Statistical Software,* **36,** 1-48.

Wilkie, E.M., Bishop, M.J. & O'Connor, W.A. (2012) Are native *Saccostrea glomerata* and invasive *Crassostrea gigas* oysters' habitat equivalents for epibenthic communities in south-eastern Australia? *Journal of Experimental Marine Biology and Ecology*, **420,** 16-25.

Wolcott, T.G. (1973) Physiological ecology and intertidal zonation in limpets (Acmaea): A critical look at" limiting factors". *The Biological Bulletin,* **145,** 389-422.

Data sources

Abdo, T. (2015) The effects of different types of concrete compositions on benthic organisms under an Ecodock. MSc, New York: New York Harbor School.

Abdus- Samad, S. (2013) The effect of the composition of concrete on biodiversity and ecology on benthic organisms. MSc, New York Harbour School.

Abdus-Samad, S. (2011) Testing the effects of different mixtures of econcrete on the recruitment of benthic organisms in the Hudson River Estuary. MSc, New York Harbour School.

Andersson, M., Berntsson, K., Jonsson, P. & Gatenholm, P. (1999) Microtextured surfaces: towards macrofouling resistant coatings. *Biofouling*, **14,** 167-178.

Arenas, F., Sánchez, I., Hawkins, S.J. & Jenkins, S.R. (2006) The invasibility of marine algal assemblages: role of functional diversity and identity. *Ecology,* **87,** 2851-2861.

Berntsson, K., Andreasson, H., Jonsson, P., Larsson, L., Ring, K., Petronis, S. & Gatenholm, P. (2000a) Reduction of barnacle recruitment on micro-textured surfaces: analysis of effective topographic characteristics and evaluation of skin friction. *Biofouling*, **16,** 245-261.

Berntsson, K.M., Jonsson, P.R., Larsson, A.I. & Holdt, S. (2004) Rejection of unsuitable substrata as a potential driver of aggregated settlement in the barnacle *Balanus improvisus*. *Marine Ecology Progress Series*, **275,** 199-210.

Berntsson, K.M., Jonsson, P.R., Lejhall, M. & Gatenholm, P. (2000b) Analysis of behavioural rejection of micro-textured surfaces and implications for recruitment by the barnacle *Balanus improvisus*. *Journal of Experimental Marine Biology and Ecology*, **251,** 59-83.

Bers, A.V. & Wahl, M. (2004) The influence of natural surface microtopographies on fouling. *Biofouling*, **20,** 43-51.

Bortone, S.A., Martin, T. & Bundrick, C.M. (1994) Factors affecting fish assemblage development on a modular artificial reef in a northern Gulf of Mexico estuary. *Bulletin of Marine Science,* **55,** 319-332.

Bourget, E., DeGuise, J. & Daigle, G. (1994) Scales of substratum heterogeneity, structural complexity, and the early establishment of a marine epibenthic community. *Journal of Experimental Marine Biology and Ecology,* **181,** 31-51.

Brotto, D.S., Krohling, W. & Zalmon, I.R. (2006) Fish community modeling agents on an artificial reef on the northern coast of Rio de Janeiro-Brazil. *Brazilian Journal of Oceanography,* **54,** 205-212.

Cacabelos, E., Martins, G.M., Thompson, R., Prestes, A.C., Azevedo, J.M.N. & Neto, A.I. (2016) Material type and roughness influence structure of inter-tidal communities on coastal defenses. *Marine Ecology,* **37,** 801-812.

Chabot, R. & Bourget, E. (1988) Influence of substratum heterogeneity and settled barnacle density on the settlement of cypris larvae. *Marine Biology,* **97,** 45-56.

Chapman, M.G. & Blockley, D.J. (2009) Engineering novel habitats on urban infrastructure to increase intertidal biodiversity. *Oecologia,* **161,** 625-635.

Chapman, M.G. & Underwood, A.J. (2011) Evaluation of ecological engineering of "armoured" shorelines to improve their value as habitat. *Journal of Experimental Marine Biology and Ecology,* **400,** 302-313.

Clark, S. & Edwards, A. (1995) Coral transplantation as an aid to reef rehabilitation: evaluation of a case study in the Maldive Islands. *Coral Reefs,* **14,** 201-213.

Clark, S. & Edwards, A.J. (1999) An evaluation of artificial reef structures as tools for marine habitat rehabilitation in the Maldives. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **9,** 5-21.

Clynick, B., Chapman, M. & Underwood, A. (2007) Effects of epibiota on assemblages of fish associated with urban structures. *Marine Ecology Progress Series,* **332,** 201-210.

Code, T. (1999) Effects of differences in hole size of artificial reef units on coral reef fish abundance and species richness. *Physis***, 1**.

Coombes, M.A., La Marca, E.C., Naylor, L.A. & Thompson, R.C. (2015) Getting into the groove: Opportunities to enhance the ecological value of hard coastal infrastructure using fine-scale surface textures. *Ecological Engineering,* **77,** 314-323.

Dafforn, K.A., Glasby, T.M. & Johnston, E.L. (2012) Comparing the invasibility of experimental reefs with field observations of natural reefs and artificial structures. *PLoS One,* **7,** e38124.

Davies, S.W., Matz, M.V. & Vize, P.D. (2013) Ecological complexity of coral recruitment processes: effects of invertebrate herbivores on coral recruitment and growth depends upon substratum properties and coral species. *PLoS One,* **8,** e72830.

Diaz-Pulido, G. & McCook, L.J. (2004) Effects of live coral, epilithic algal communities and substrate type on algal recruitment. *Coral Reefs,* **23,** 225-233.

Dobretsov, S.V. & Railkin, A.I. (1996) Effects of substrate features on settling and attachment of larvae in blue mussel *Mytilus edulis* (Mollusca, Filibranchia). *Zoologicheskii Zhurnal,* **75,** 499-506.

Dudgeon, S. & Petraitis, P.S. (2005) First year demography of the foundation species, Ascophyllum nodosum, and its community implications. *Oikos,* **109,** 405-415.

Dugan, J.E., Airoldi, L., Chapman, M.G., Walker, S.J., Schlacher, T., Wolanski, E. & McLusky, D. (2011) 8.02 - Estuarine and coastal structures: environmental effects, a focus on shore and nearshore structures. *Treatise on Estuarine and Coastal Science,* pp. 17-41. Academic Press, Waltham.

Edmunds, P.J., Nozawa, Y. & Villanueva, R.D. (2014) Refuges modulate coral recruitment in the Caribbean and the Pacific. *Journal of Experimental Marine Biology and Ecology,* **454,** 78-84.

Edwards, A.J., Guest, J.R., Heyward, A.J., Villanueva, R.D., Baria, M.V., Bollozos, I.S. & Golbuu, Y. (2015) Direct seeding of mass-cultured coral larvae is not an effective option for reef rehabilitation. *Marine Ecology Progress Series,* **525,** 105-116.

Evans, A., Firth, L., Hawkins, S., Morris, E., Goudge, H. & Moore, P. (2015) Drill-cored rock pools: an effective method of ecological enhancement on artificial structures. *Marine and Freshwater Research,* **67,** 123-130.

Evans, A.J. (2016) Artificial coastal defence structures as surrogate habitats for natural rocky shores: giving nature a helping hand. Aberystwyth University.

Fernández, T.V., D'Anna, G., Badalamenti, F. & Pérez-Ruzafa, A. (2009) Effect of simulated macroalgae on the fish assemblage associated with a temperate reef system. *Journal of Experimental Marine Biology and Ecology,* **376,** 7-16.

Ferse, S.C.A. (2008) Artificial reef structures and coral transplantation: fish community responses and effects on coral recruitment in North Sulawesi/Indonesia. Dissertation. University of Bremen, Bremen, Germany.

Ferse, S.C.A., Nugues, M.M., Romatzki, S.B.C. & Kunzmann, A. (2013) Examining the use of mass transplantation of brooding and spawning corals to support natural coral recruitment in Sulawesi/Indonesia. *Restoration Ecology,* **21,** 745-754.

Firth, L.B., Schofield, M., White, F.J., Skov, M.W. & Hawkins, S.J. (2014) Biodiversity in intertidal rock pools: Informing engineering criteria for artificial habitat enhancement in the built environment. *Marine Environmental Research,* **102,** 122-130.

Goff, M. (2010) Evaluating habitat enhancements of an urban intertidal seawall: Ecological responses and management implications. MSc, University of Washington.

Gorham, J.C. & Alevizon, W.S. (1989) Habitat complexity and the abundance of juvenile fishes residing on small scale artificial reefs. *Bulletin of Marine Science,* **44,** 662-665.

Gratwicke, B. & Speight, M.R. (2005) Effects of habitat complexity on Caribbean marine fish assemblages. *Marine Ecology Progress Series,* **292,** 301-310.

Guarnieri, G., Terlizzi, A., Bevilacqua, S. & Fraschetti, S. (2009) Local vs regional effects of substratum on early colonization stages of sessile assemblages. *Biofouling,* **25,** 593-604.

Hair, C. & Bell, J. (1992) Effects of enhancing pontoons on abundance of fish: initial experiments in estuaries. *Bulletin of Marine Science,* **51,** 30-36.

Harlin, M.M. & Lindbergh, J.M. (1977) Selection of substrata by seaweeds: optimal surface relief. *Marine Biology,* **40,** 33-40.

Hawkins, S.T. (1998) The epibenthic colonization of artificial subtidal habitats at the Cape d'Aguilar Marine Reserve, Hong Kong. PhD University of Hong Kong.

Heath, T. & Moody, G. (2003) Habitat development along a highly urbanised foreshore. *Coastal Conference*, pp. 1-7.

Heyward, A., Smith, L., Rees, M. & Field, S. (2002) Enhancement of coral recruitment by in situ mass culture of coral larvae. *Marine Ecology Progress Series,* **230,** 113-118.

Hills, J., Thomason, J. & Muhl, J. (1999) Settlement of barnacle larvae is governed by Euclidean and not fractal surface characteristics. *Functional Ecology,* **13,** 868-875.

Hixon, M.A. & Beets, J.P. (1989) Shelter characteristics and Caribbean fish assemblages: experiments with artificial reefs. *Bulletin of Marine Science,* **44,** 666-680.

Hixon, M.A. & Beets, J.P. (1993) Predation, prey refuges, and the structure of coral-reef fish assemblages. *Ecological Monographs,* **63,** 77-101.

Hunter, W.R. & Sayer, M.D.J. (2009) The comparative effects of habitat complexity on faunal assemblages of northern temperate artificial and natural reefs. *ICES Journal of Marine Science: Journal du Conseil***,** fsp058.

Kellison, T.G. & Sedberry, G.R. (1998) The effects of artificial reef vertical profile and hole diameter on fishes off South Carolina. *Bulletin of Marine Science,* **62,** 763-780.

Köhler, J., Hansen, P. & Wahl, M. (1999) Colonization patterns at the substratum-water interface: How does surface microtopography influence recruitment patterns of sessile organisms? *Biofouling,* **14,** 237-248.

Langhamer, O. & Wilhelmsson, D. (2009) Colonisation of fish and crabs of wave energy foundations and the effects of manufactured holes a field experiment. *Marine Environmental Research,* **68,** 151-157.

Lapointe, L. & Bourget, E. (1999) Influence of substratum heterogeneity scales and complexity on a temperate epibenthic marine community. *Marine Ecology Progress Series,* **189,** 159-170.

Lemire, M. & Bourget, E. (1997) Substratum heterogeneity and complexity influence micro-habitat selection of Balanus sp. and Tubularia crocea larvae. *Oceanographic Literature Review,* **1,** 48.

Lingo, M.E. & Szedlmayer, S.T. (2006) The influence of habitat complexity on reef fish communities in the northeastern Gulf of Mexico. *Environmental Biology of Fishes,* **76,** 71-80.

Loke, L.H., Liao, L.M., Bouma, T.J. & Todd, P.A. (2016) Succession of seawall algal communities on artificial substrates. *Raffles Bulletin of Zoology,* **32,** 1-10.

Loke, L.H. & Todd, P.A. (2016) Structural complexity and component type increase intertidal biodiversity independently of area. *Ecology,* **97,** 383-393.

Margiotta, A.M., Shervette, V.R., Hadley, N.H., Plante, C.J. & Wilber, D.H. (2016) Species-specific responses of resident crabs to vertical habitat complexity on intertidal oyster reefs. *Journal of Experimental Marine Biology and Ecology,* **477,** 7-13.

Martins, G.M., Jenkins, S.R., Neto, A.I., Hawkins, S.J. & Thompson, R.C. (2016) Long-term modifications of coastal defences enhance marine biodiversity. *Environmental Conservation,* **43,** 109-116.

Morris, R.L. (2016) Retrofitting biodiversity: Ecological engineering for management of urbanised systems. PhD, Univeristy of Sydney.

Moschella, P., Abbiati, M., Åberg, P., Airoldi, L., Anderson, J., Bacchiocchi, F., Bulleri, F., Dinesen, G.E., Frost, M. & Gacia, E. (2005) Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. *Coastal Engineering,* **52,** 1053-1071.

Nozawa, Y., Tanaka, K. & Reimer, J.D. (2011) Reconsideration of the surface structure of settlement plates used in coral recruitment studies. *Zoological Studies,* **50,** 53-60.

Ogata, E. (1953) Some experiments on the settling of spores of red algae. *Bulletin of Society of Plant Ecology Tokoyo,* **2,** 104107.

Paalvast, P. (2015) The role of geometric structure and texture on concrete for algal and macrofaunal colonization in the marine and estuarine intertidal zone. *Proceedings of RECIF Conference on artificial reefs : from materials to ecosystems*. Caen, France.

Pech, D., Ardisson, P.L. & Bourget, E. (2002) Settlement of a tropical marine epibenthic assemblage on artificial panels: influence of substratum heterogeneity and complexity scales. *Estuarine Coastal and Shelf Science*, **55,** 743-750.

Perkol-Finkel, S., Ferrario, F., Nicotera, V. & Airoldi, L. (2012) Conservation challenges in urban seascapes: promoting the growth of threatened species on coastal infrastructures. *Journal of Applied Ecology,* **49,** 1457-1466.

Perkol-Finkel, S. & Sella, I. (2014) Ecologically active concrete for coastal and marine infrastructure: innovative matrices and designs. *Proceeding of the 10th ICE Conference: from Sea to Shore–Meeting the Challenges of the Sea. ICE Publishing, 18–20 September 2013-Edinburgh, UK*, pp. 1139-1150.

Pinn, E.H., Mitchell, K. & Corkill, J. (2005) The assemblages of groynes in relation to substratum age, aspect and microhabitat. *Estuarine, Coastal and Shelf Science,* **62,** 271-282.

Pomerat, C. & Weiss, C. (1946) The influence of texture and composition of surface on the attachment of sedentary marine organisms. *The Biological Bulletin,* **91,** 57-65.

Quinn, T.P. (2009) The influence of artificial reef associated fish assemblages and varying substrates on coral recruitment. PhD. Nova Southeastern University

Reed, D.C., Schroeter, S.C. & Huang, D.J.-S. (2006) *An experimental investigation of the use of artificial reefs to mitigate the loss of giant kelp forest habitat: A case study of the San Onofre nuclear generating station's artificial reef project*. Sea Grant Extension Program, UC Cooperative Extension. 143 pp.

Rilov, G. & Benayahu, Y. (1998) Vertical artificial structures as an alternative habitat for coral reef fishes in disturbed environments. *Marine Environmental Research,* **45,** 431-451.

Rountree, R.A. (1990) Community structure of fishes attracted to shallow water fish aggregation devices off South Carolina, USA. *Environmental Biology of Fishes,* **29,** 241-262.

Savoya, V. & Schwindt, E. (2010) Effect of the substratum in the recruitment and survival of the introduced barnacle *Balanus glandula* (Darwin 1854) in Patagonia, Argentina. *Journal of Experimental Marine Biology and Ecology,* **382,** 125-130.

Sellheim, K., Stachowicz, J.J. & Coates, R.C. (2009) Effects of a nonnative habitat-forming species on mobile and sessile epifaunal communities. *Marine Ecology Progress Series*, **398,** 69.

Sherman, R.L., Gilliam, D.S. & Spieler, R.E. (2002) Artificial reef design: void space, complexity, and attractants. *ICES Journal of Marine Science: Journal du Conseil*, **59,** S196-S200.

Sherrard, T.R., Hawkins, S.J., Barfield, P., Kitou, M., Bray, S. & Osborne, P.E. (2016) Hidden biodiversity in cryptic habitats provided by porous coastal defence structures. *Coastal Engineering*, **118,** 12-20.

Skinner, L.F. & Coutinho, R. (2005) Effect of microhabitat distribution and substrate roughness on barnacle *Tetraclita stalactifera* (Lamarck, 1818) settlement. *Brazilian Archives of Biology and Technology*, **48,** 109-113.

Smith, R.S., Johnston, E.L. & Clark, G.F. (2014) The role of habitat complexity in community development is mediated by resource availability. *PLoS One*, **9,** e102920.

Soniat, T.M., Finelli, C.M. & Ruiz, J.T. (2004) Vertical structure and predator refuge mediate oyster reef development and community dynamics. *Journal of Experimental Marine Biology and Ecology*, **310,** 163-182.

Thomason, J.C., Letissier, M.D.A.A., Thomason, P.O. & Field, S.N. (2002) Optimising settlement tiles: the effects of surface texture and energy, orientation and deployment duration upon the fouling community. *Biofouling*, **18,** 293-304.

Van Tamelen, P.G., Stekoll, M.S. & Deysher, L. (1997) Recovery processes of the brown alga Fucus gardneri following the'Exxon Valdez' oil spill: Settlement and recruitment. *Marine Ecology Progress Series*, **160,** 265-277.

Vucko, M., Poole, A., Carl, C., Sexton, B., Glenn, F., Whalan, S. & de Nys, R. (2014) Using textured PDMS to prevent settlement and enhance release of marine fouling organisms. *Biofouling*, **30,** 1-16.

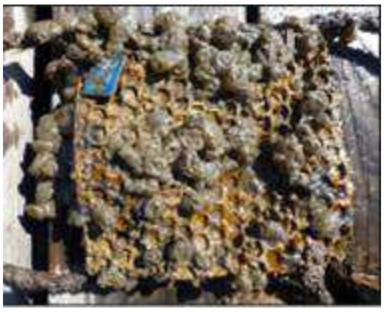
Walters, L.J. & Wethey, D.S. (1996) Settlement and early post-settlement survival of sessile marine invertebrates on topographically complex surfaces: the importance of refuge dimensions and adult morphology. *Marine Ecology Progress Series*, **137,** 161.

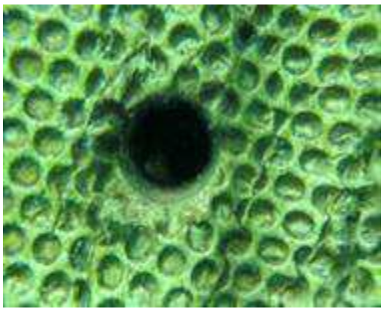
Watanuki, A. & Yamamoto, H. (1990) Settlement of seaweeds on coastal structures. *Thirteenth International Seaweed Symposium*, pp. 275-280. Springer.

Wilhelmsson, D., Yahya, S.A.S. & Ohman, M.C. (2006) Effects of high-relief structures on cold temperate fish assemblages: A field experiment. *Marine Biology Research*, **2,** 136-147.

Wilkie, E.M., Bishop, M.J. & O'Connor, W.A. (2012) Are native *Saccostrea glomerata* and invasive *Crassostrea gigas* oysters' habitat equivalents for epibenthic communities in south-eastern Australia? *Journal of Experimental Marine Biology and Ecology*, **420,** 16-25.

Table 1: Categories of intervention defined for the meta-analysis and qualitative literature review.

Classification	**Description**	**Image**
Texture	micro-scale manipulation applied to an entire intertidal or subtidal surface that produces depressions and/or raises of ≤1 mm	
Crevice	intertidal or subtidal depression with a length to width ratio >3:1, and depth of >1 mm	
Pit	intertidal or subtidal depressions with a length to width ratio <3:1 and depth of >1 mm to 5 cm. This may or may not hold water.	
Intertidal water retaining features	intertidal depressions or features including a) flower pots and b) rockpools with a length to width ratio <3:1 that hold water (≥5 cm depth) when the tide retreats	a) flowerpot  b) rockpools

		
Subtidal holes	subtidal depressions with a length to width ratio <3:1 and ≥5 cm depth	
Small elevations	intertidal or subtidal protruding structures (i.e. raises, ledges or ridges) ≥ 1 mm high and < 0.5 m high in dimension	
Large elevations	intertidal or subtidal protruding structures (i.e. raises, ledges, ridges) > 0.5 m high in dimension	
Soft structures	subtidal flexible, protruding materials such as rope, ribbon or twine (>0.1 m in length)	
Habitat-forming	taxa that provide structural habitat to	

taxa	associated organisms (i.e. barnacles, bivalves, coral, canopy-forming algae, branching coralline algae)	

Table 2: Outcome of meta-analyses and qualitative (underlined and in brackets) review. For each intervention we highlight the results of the meta-analysis where available and the results from the qualitative review where there was insufficient information presented to undertake the meta-analysis (see figures for full results). Interventions are scored according to whether they had significant positive (+), negative (-) or non-significant (ns) effects (at $\alpha = 0.05$) relative to controls.

Response	Number of species				Abundance of species				Number of species or abundance of habitat-forming taxa				
Microhabitat	Sessile	Mobile	Benthic	Fish	Sessile	Mobile	Benthic	Fish	Barnacles	Bivalves	Branching coralline	Canopy algae	Coral
Intertidal													
Texture	(ns)				+				+	-	+	(ns)	
Crevice	+	ns	ns		+	(+)	ns		+	+		ns	
Pit			+			(+)	ns		+	(+)	(+)	ns	
Small elevation			ns				ns		(+)	ns		(+)	
Water-retaining	+	ns	+	(+)					ns	ns	+	+	
Seeding	+	ns			ns	(ns)				ns			
Subtidal													
Texture	(ns)				+				-	+	+		(ns)

Crevice	ns				ns				ns	ns		ns	
Pit									<u>(ns)</u>				+
Hole				ns		+		ns					
Large elevation				ns				+					
Seeding	ns			+	ns	ns		ns		ns	ns	ns	ns
Soft structure				+				+					

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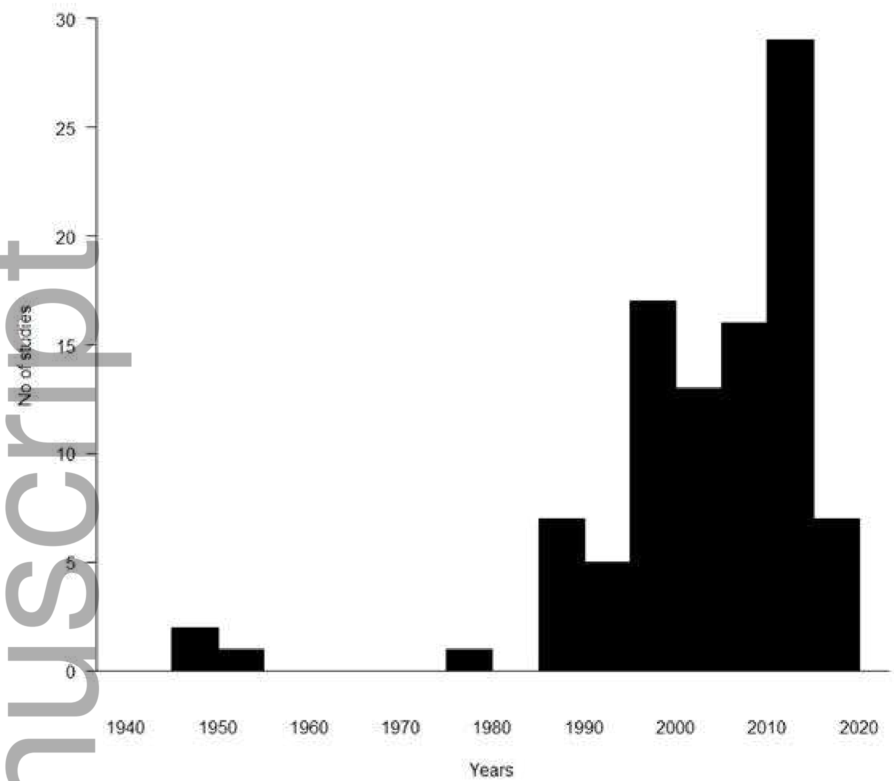

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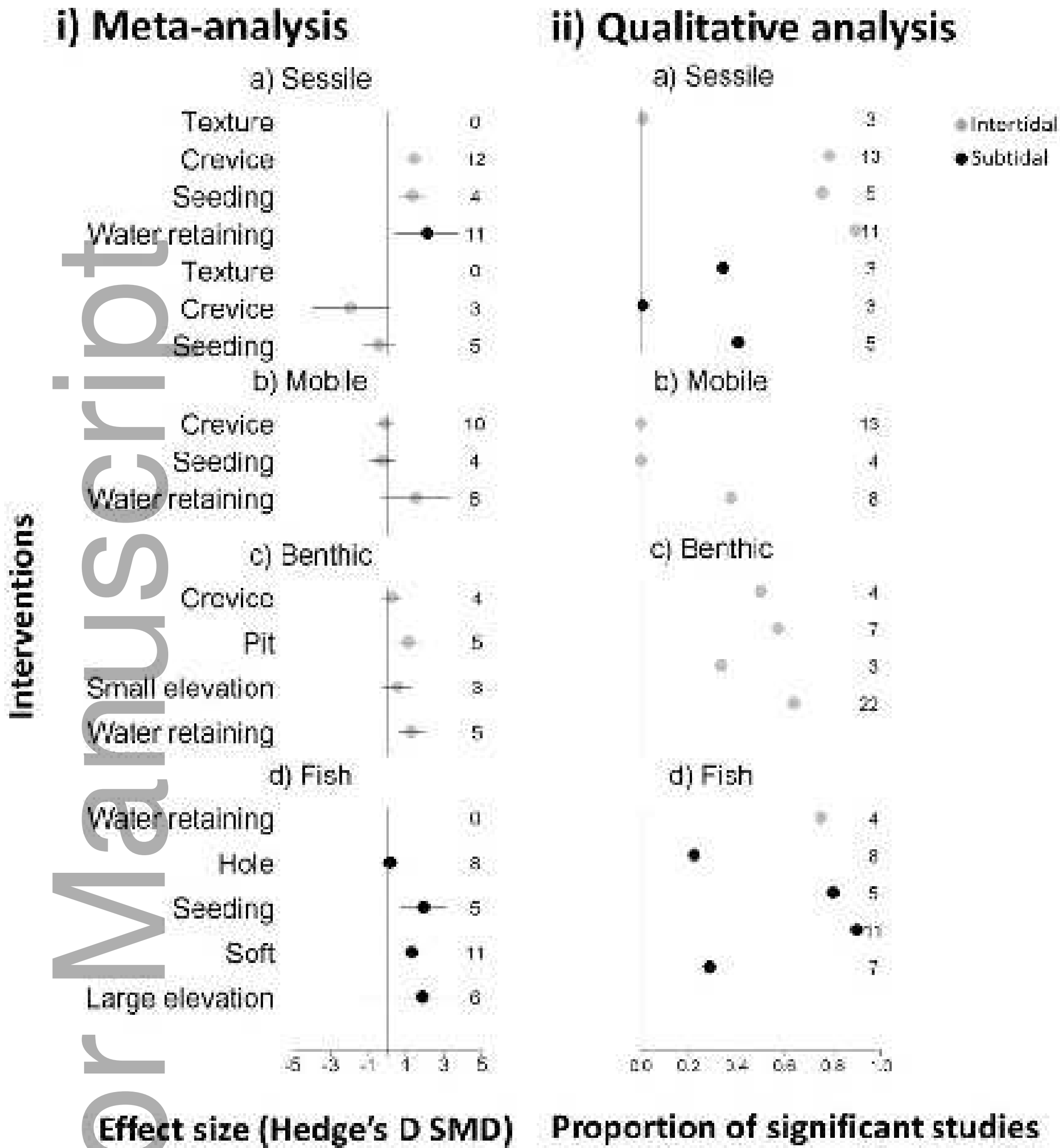

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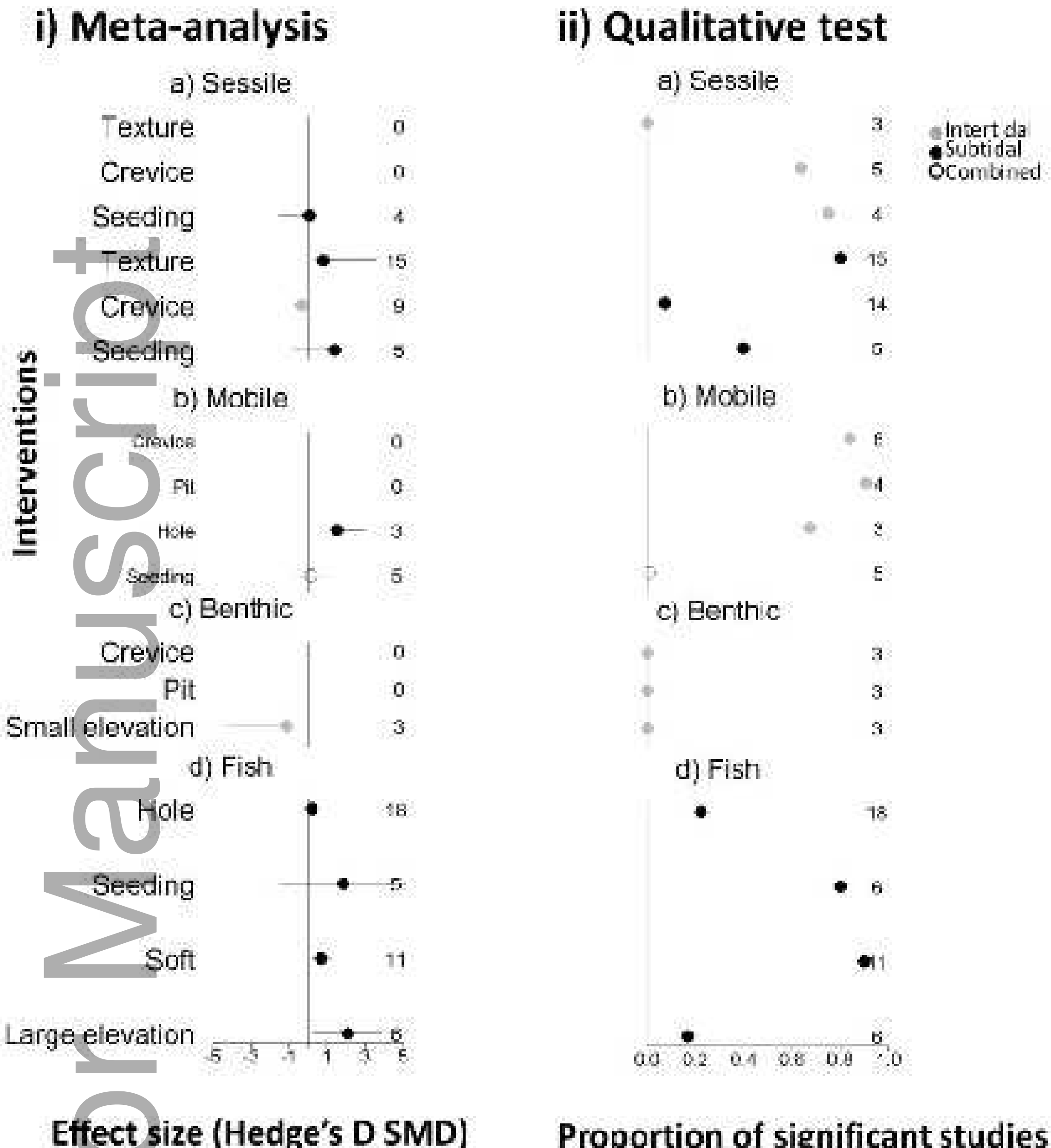

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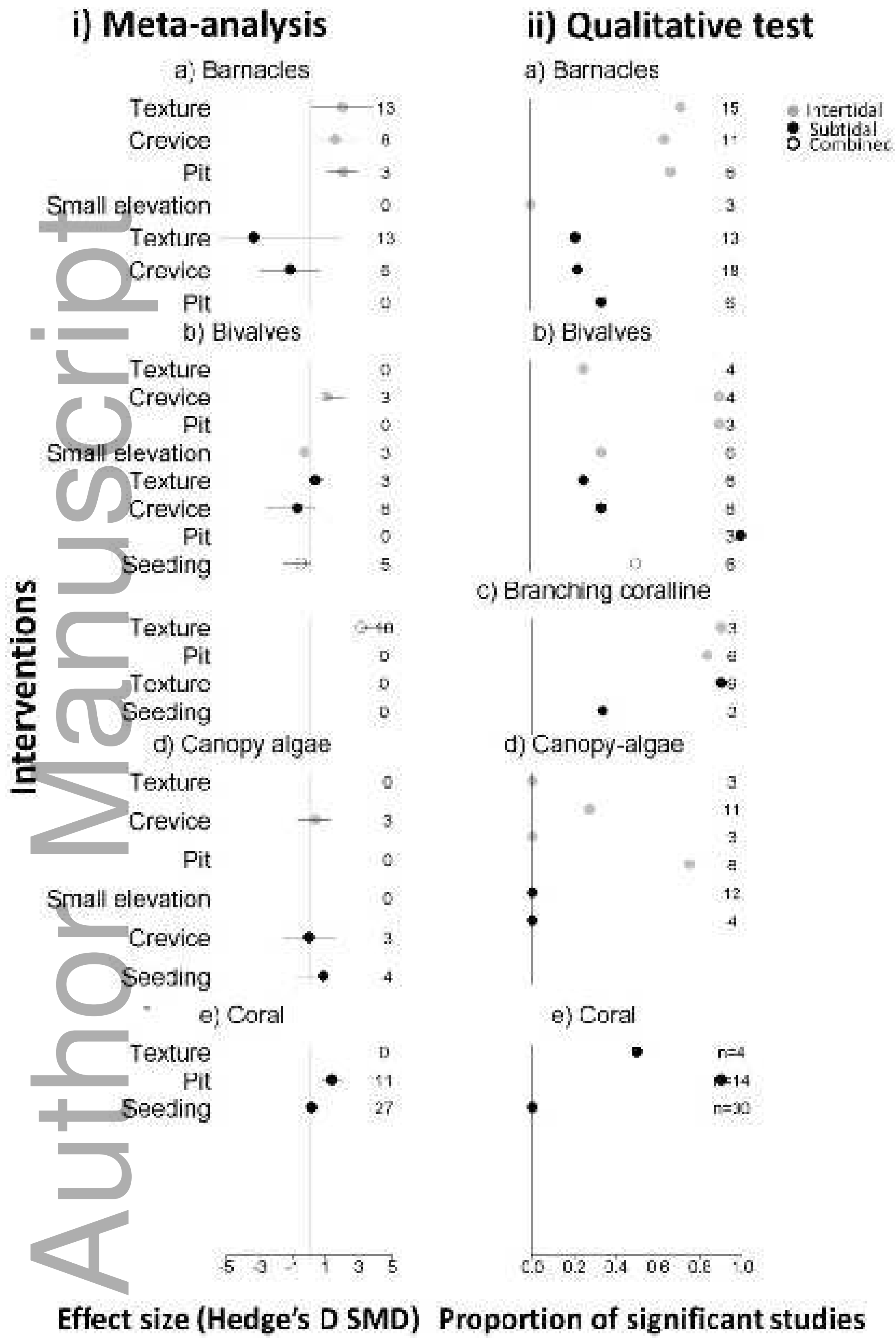

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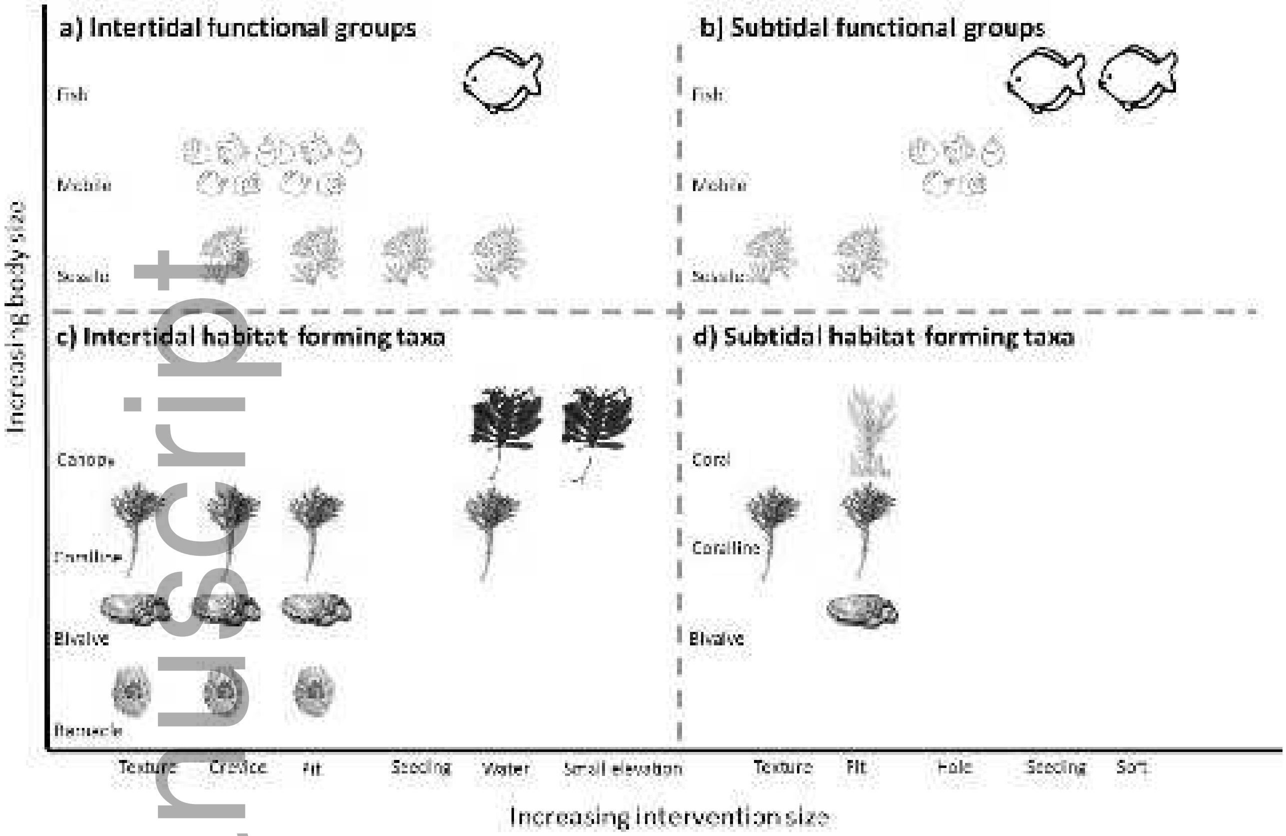

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